FINAL REPORT

Title: Quantifying the effects of post-fire decision-making on forest recovery in a severely burned southwestern landscape

JFSP PROJECT ID:16-1-05-8

November 2020

Matthew D. Hurteau

Department of Biology, University of New Mexico

Dan J. Krofcheck

Department of Biology, University of New Mexico

Craig D. Allen

US Geological Survey, New Mexico Lands Field Station

FIRESCIENCE.GOV
Research Supporting Sound Decisions

Table of Contents

List of Abbreviations/Acronyms

AIC – Akaike Information Criterion

CWB – Climatic Water Balance

DEM – Digital Elevation Model

LANDIS-II – Landscape Disturbance and Succession Model

PPT – Precipitation

VPD – Vapor Pressure Deficit

WY – Water Year

Keywords

Post-fire, Regeneration, High-severity, Management

Acknowledgements

We wish to acknowledge Cécile Remy and Alisa Keyser for their work on the simulation modeling components of this project. We wish to thank Briana Becerra, Joe Crockett, Rachel Swanteson-Franz, and Kevin Willson for their help with field work.

Abstract

Fire-exclusion and changing climatic conditions are increasing the area burned and the size of high-severity burn patches in historically frequent-fire forests. In the southwestern US, distance to seed source and hot, dry conditions in high-severity patches is limiting tree regeneration. While post-fire tree planting can overcome dispersal limitations, high planted seedling mortality rates are common in the southwest. Microclimatic conditions are influential on tree seedling survival and can vary as a function of topography and vegetation cover type. We sought to determine how planted seedling survival and growth would vary as a function of aspect and vegetation cover type in the footprint of the 2011 Las Conchas Fire in northern New Mexico. We also sought to determine how the probability of fire would influence landscape successional development. To quantify the effects of aspect and cover type on seedling survival and growth, we planted ponderosa pine and Douglas-fir seedlings by aspect (north vs south) and cover type (open, under shrub) in a full factorial design. To quantify the effects of fire probability on landscape successional development, we used a simulation model to test the sensitivity of the system to increasing fire probability. We found that topographic wetness index and topographic roughness index were good predictors of both survival and growth. We also found that increasing fire probability can prevent the establishment of tree seedlings following high-severity wildfire, causing the transition from forest to non-forest to be reinforced.

Objectives

Fire exclusion in the dry forests of the western US has created homogenous landscapes of continuous forest canopy and fuels (Fulé et al. 1997, North et al. 2007). When these conditions are coupled with climate-driven increases in wildfire size, the landscape is increasingly characterized by large patches of homogenous fire severity; this combination results in large areas of uniform post-fire vegetation composition (Miller et al. 2009, Westerling et al. 2011, Collins and Roller 2013, Singleton et al. 2019). Large areas of high severity fire can transition the landscape to a non-forested condition and these post-fire outcomes can be further reinforced by subsequent burning. The severity of a second wildfire at a given location is correlated with fire severity during the initial wildfire (Coppoletta et al. 2016, Holden et al. 2010) and subsequent high severity fire can reinforce the non-forested condition (Coop et al.2016).

Natural post-fire plant community development can both facilitate tree seedling establishment by buffering climatic conditions and impede tree seedling establishment by serving as a fuel source for subsequent fires (Gomez-Aparicio et al. 2005, McGinnis et al. 2010). Understanding how post-fire revegetation efforts interact with natural post-fire plant establishment is central to effectively deploying scarce resources in landscapes increasingly impacted by large, hot fires. To identify effective post-fire landscape management options, we conducted a field experiment to improve our understanding of planted seedling establishment in severely burned landscapes and ran simulations to model post-fire landscape successional trajectories.

Background

In the southwestern US, post-fire plant community development coupled with climatic conditions that are more conducive to burning result in a relatively short window where a recently burned area is effective at limiting subsequent fire spread (Parks et al. 2015). Coupled with the reinforced non-forested condition that follows with subsequent burning, this short post-burn window presents a challenge for managers as they attempt to re-forest large, high-severity burn patches. Increasing temperature and high interannual precipitation variability further exacerbate this management challenge. These climate factors heavily influence post-fire seedling success rates in the near-term (1-2 years, Savage et al. 2013) and planting success is dependent on a fire-free period for seedlings to achieve a size class sufficient to survive subsequent burning.

Part of the challenge of reforesting severely burned sites is that the post-fire landscape is characterized by increased temperature and lowered relative humidity, conditions that are nearly always stressful and often lethal to planted seedlings. Naturally establishing vegetation, such as shrubs, may provide a microclimatic buffer that facilitates planted seedling establishment. The buffering effect of shrub canopy shading has been shown to facilitate tree seedling establishment in Mediterranean systems (Gomez-Aparicio et al. 2005). Yet, there is also evidence that demonstrates competition for water between shrubs and seedlings, which can impede establishment (Plamboeck et al. 2008). Interspecific interactions can change and the potential exists for a transition from facilitation of seedling establishment by shrubs modifying the microclimate to competition with shrubs for light and water (Holmgren et al. 1997, Sthultz et al. 2007). These microclimatic effects may be further enhanced on hill slopes with north facing aspects, where reduced solar insolation lowers the evaporative demand from the atmosphere, resulting in greater moisture availability relative to south facing slopes (e.g., Geroy et al. 2011). Furthermore, shrubs also serve as a fuel source for subsequent burning and the fire intensity from burning shrubs can be lethal for establishing trees for 20 years following planting (McGinnis et al. 2010). Thus, the efficacy of shrubs as a facilitator of planted seedling establishment may be limited by both the competitive interaction for water and the shrubs contribution to creating a continuous fuel layer. Better understanding of the role of shrubs in buffering microclimate and their contribution to fire risk for seedlings is central to projecting post-fire forest development and identifying post-fire management strategies that will improve re-forestation efforts.

Materials and Methods

We used a combination of field experiments and simulation modeling to answer the following questions: 1) How does post-fire vegetation community type modify microclimate and influence seedling survival rate?; 2) What combination of post-fire vegetation community, edaphic, and topographic conditions yields the highest seedling survival rates?; 3) How does post-fire vegetation community type alter the fire-free time interval required to increase planted tree survivorship during a re-burn?; 4) How do these factors alter the forest recovery trajectory across the landscape?

We chose the footprint of the Las Conchas fire on the east flank of the Jemez Mountains in northern New Mexico as our study location because it burned across US Forest Service, National Park Service and tribal lands, exhibited extreme fire behavior, and burned through vegetation ranging from juniper woodlands at the lowest elevations to mixed-conifer forest at the highest elevations (Fig. 1). The Las Conchas fire also re-burned portions of three previous wildfires.

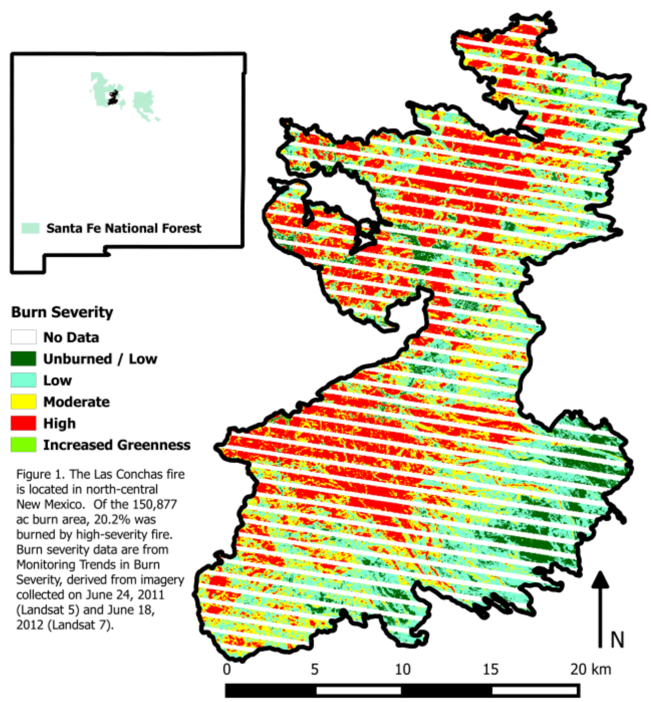

Figure 1. The Las Conchas fire is located in north-central New Mexico. Of the 150,877 ac burn area, 20.2% was burned by high-severity fire. Burn severity data are from Monitoring Trends in Burn Severity, derived from imagery collected on June 24, 2011 (Landsat 5) and June 18, 2012 (Landsat 7).

We planted 200 seedlings of Douglas-fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) stratified by aspect (Moist [NW, N, NE] and Dry [SW, S, SE]) and post-fire vegetation patch (shrub and grass). We measured measure temperature and relative humidity using iButton sensors during the growing season. We also installed weather stations to measure common meteorological variables and soil moisture and soil temperature in both shrub and grass patches. We measured seedling basal diameter and height at the start and end of each growing season and conducted monthly growing season tallies of live/dead status during the first two growing seasons. Seedlings were planted during fall 2016.

We used ibutton and weather station data to calculate a range of summary meteorological variables and we calculated a range of topographic variables using USGS DEM data. We analyzed seedling survival using logistic regression to quantify the contribution of each of the predictor variables to explaining variability in survival. Prior to performing analyses, we develop a set of predictor variables that included both topographic and meteorological variables. We ran model selection procedures for topographic variable only models to evaluate the effects of topography on survival in isolation of meteorological variables. Prior to running model selection procedures on models that included topographic and meteorological variables, we ran single predictor logistic regressions. We carried forward variables that explained at least 1% of the variability. We modeled seedling growth using linear models to predict growth as a function of topographic indices. We included elevation, heat load, topographic roughness index, and topographic wetness index, and log transformed values of each predictor to model ponderosa pine and Douglas-fir height. We performed statistical analyses in R using the MuMIn package (R Core Team 2019, Bartoń 2020).

To run simulations of the landscape, we used the landscape disturbance and succession simulation model LANDIS-II to simulate vegetation dynamics (Scheller et al. 2011). Two extensions to the model, PnET Succession and Dynamic Fire and Fuels, provide the capacity to simulate changing climate and fire frequency (Scheller et al. 2011, de Bruijn et al. 2014). The PnET Succession extension is based on the PnET ecophysiology model and uses species-specific parameters, soil characteristics, and climate to simulate carbon and nitrogen dynamics that influence vegetation change. The Dynamic Fire and Fuels extension is based on the Canadian Fire Prediction System and uses spread equations from Finney (2002). Fire behavior is determined by weather, topography, and fuels. This extension provides the tools necessary to simulate a range of fire characteristics, including altering ignition probabilities and fire size distributions. As part of this project, we parameterized and validated the model for southwestern forest types (Remy et al. 2019). To quantify the effects of fire on landscape development, we ran a simulation experiment that included three different ignition probabilities. The simulation experiment is described in detail in Keyser et al. (2020).

Results and Discussion

Overall seedling survival was higher for ponderosa pine (25.5%) than for Douglas-fir (9.8%) after three growing seasons. Models that included only topographic variables explained 17% of the variability in survival for ponderosa pine and 20% for Douglas-fir (Tables 1, 2). The three best ponderosa pine survival models all included topographic roughness index, slope and topographic wetness index (Table 1). The positive relationship between survival and topographic roughness and wetness indicates that ponderosa pine survival was higher in areas that had increased topographic complexity and in areas of the landscape that tend to accumulate water. The best Douglas-fir survival model included all topographic predictor variables (Table 2). Similar to ponderosa pine, more topographically complex areas and areas that accumulate water had higher Douglas-fir survival. While the Douglas-fir model did have a negative relationship with elevation, our plots only differed by 195 m in elevation.

Table 1 The five logistic models with the lowest ΔAICc for ponderosa pine three-year survival using only topographic predictor variables.

Intercept	Elevation	Heat Load	Topographic Roughness	Slope	Topographic Index	Topographic Wetness	R2	AICc	ΔAICc
-6.94	-	-	29.14	-3.10	-	0.83	0.170	190.43	0
-12.72	-	-	29.74	-3.18	0.005	0.75	0.173	191.91	1.47
-12.71	0.002	-	29.74	-3.18	-	0.75	0.173	191.92	1.48
-6.81	-	-0.16	28.96	-3.08	-	0.83	0.170	192.53	2.09
-15.30	-1.94	-	31.95	-3.42	4.05	0.78	0.175	193.43	2.99

Table 2 The three best logistic models of Douglas-fir three-year survival using only topographic predictor variables.

Intercept	Elevation	Heat Load	Topographic Roughness	Slope	Topographic Index	Topographic Wetness	R2	AICc	ΔAICc
-69.42	-20.65	8.50	69.28	-7.45	43.06	1.15	0.209	93.42	0
-64.66	-16.16	5.59	69.15	-7.57	33.73	-	0.185	96.9	3.47
-42.469	-10.37	-	43.69	-4.68	21.65	-	0.166	99.21	5.78

When evaluating three-year survival models that included both topographic and meteorological predictor variables, we were able to explain 32% of the variability in ponderosa pine survival and 31% in Douglas-fir survival (Tables 3, 4). The meteorological variables that were consistent across the best ponderosa pine three-year survival models included: water year 1 (WY1) spring cumulative climatic water balance (CWB), WY1 total spring precipitation, spring mean daily maximum vapor pressure deficit (VPD), and WY2 spring CWB (Table 3). There were 28 models within $\Delta AIC_c = 2$ of the best model. We only present the five lowest AIC_c models because the common predictors were common across all of the models within $\Delta AIC_c = 2$ of the best model.

Table 3 The five logistic models with the lowest ΔAICc for ponderosa pine three-year survival using topographic and meteorological predictor variables. CWB is climatic water balance (mm), Dry Days is the number of days between the last winter precipitation event and the first monsoon precipitation event. Mean VPD is the mean over the season and max VPD is the mean of daily maximum values over the season.

	Model 1	Model 2	Model 3	Model 4	Model 5
Intercept	26.46	35.26	12.12	12.12	30.19
Elevation	-	-	0.007	-	-
Heat Load	-	-	-	-	-
Topographic Roughness	50.35	48.89	50.46	50.46	47.43
Slope	-5.81	-5.66	-5.84	-5.84	-5.49
Topographic Index	-	-	-	0.01	-
Topographic Wetness	1.16	1.26	1.06	1.06	1.29
WY1 Spring CWB	0.23	0.27	0.29	0.29	0.26
WY1 Summer CWB	-	-	-	-	-
WY1 Dry Days	-	-0.01	-	-	-
WY1 Spring PPT	-0.29	-0.34	-0.35	-0.35	-0.33
WY1 Spring max VPD	-0.51	-0.52	-0.47	-0.47	-0.45
WY1 Summer PPT	-	-	-	-	-
WY1 Winter PPT	-	-	-	-	-
WY2 Spring CWB	-0.18	-0.21	-0.24	-0.24	-0.20
WY2 Summer CWB	-	-	-	-	-
WY2 Dry Days	-	-	-	-	0.01
R2	0.322	0.329	0.328	0.328	0.328
AICc	160.08	160.33	160.49	160.49	160.61
ΔAICc	0	0.250	0.409	0.413	0.530

Table 4 The five logistic models with the lowest ΔAICc for Douglas-fir three-year survival using topographic and meteorological predictor variables. CWB is climatic water balance (mm), Dry Days is the number of days between the last winter precipitation event and the first monsoon

precipitation event. Mean VPD is the mean over the season and max VPD is the mean of daily maximum values over the season.

	Model 1	Model 2	Model 3	Model 4	Model 5
Intercept	25994.49	16583.12	16561.27	20891.94	19501.84
Elevation	-16.94	-	0.72	-	0.86
Heat Load	-	-	-	-	-
Topographic Roughness	246.66	195.51	196.29	199.32	188.94
Slope	-24.53	-19.67	-19.75	-19.74	-18.73
Topographic Index	37.81	1.52	-	1.94	-
Topographic Wetness	-	-	-	-	-
WY1 Spring CWB	601.91	382.82	382.13	482.59	450.44
WY1 Summer CWB	-	-	-	-	-
WY1 Dry Days	9.54	6.05	6.03	7.66	7.15
WY1 Spring PPT	-625.66	-397.65	-396.96	-501.43	-467.99
WY1 Summer mean VPD	-	-3.70	-3.85	-	-
WY1 Summer VPD max	-	-	-	-0.95	-1.00
WY2 Spring CWB	-558.98	-355.63	-354.99	-448.24	-418.40
WY2 Summer CWB	70.69	45.02	44.95	56.72	52.94
WY1 Dry Days	22.74	14.38	14.35	18.19	16.97
R2	0.313	0.312	0.312	0.311	0.310
AICc	75.06	75.46	75.50	75.84	75.95
ΔAICc	0	0.40	0.44	0.78	0.88

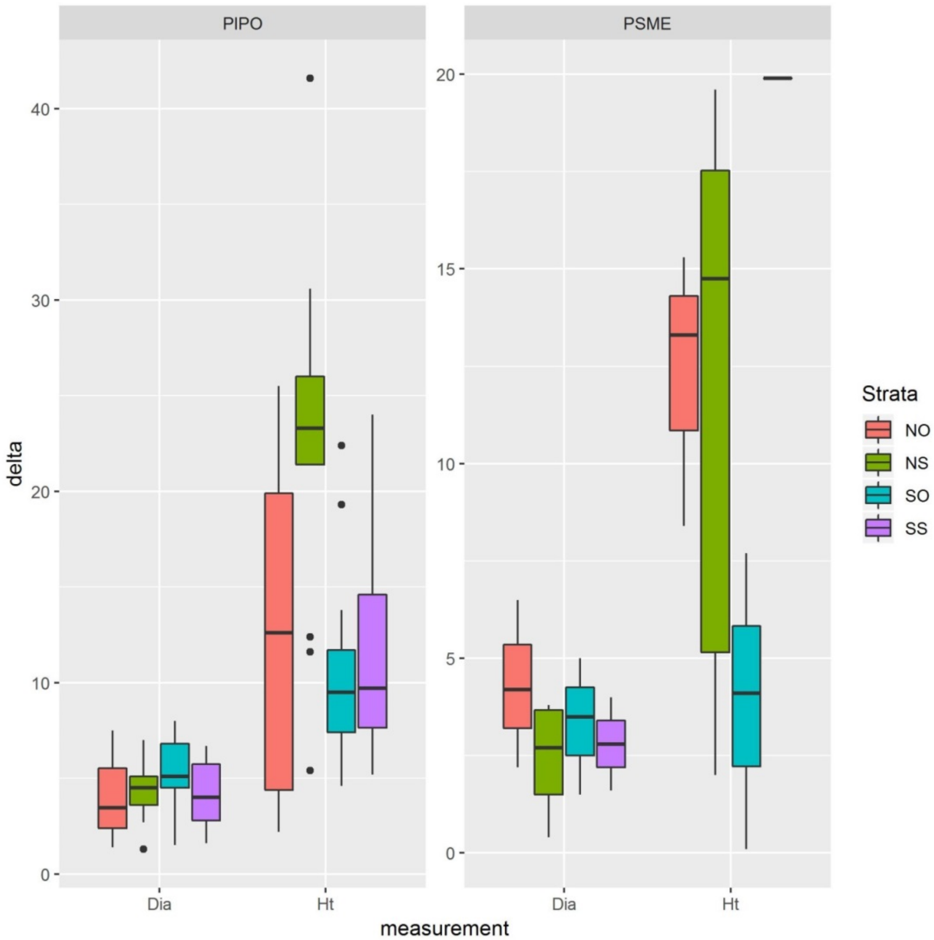

Figure 2: Distributions of change in basal diameter (Dia) and height (Ht) over three growing seasons for ponderosa pine (PIPO) and Douglas-fir (PSME). The different strata are combinations of north (N) and south (S) aspect and under shrub cover (S) or in the open (O).

We evaluated differences in basal diameter and height over the three growing seasons and found that the largest differences between treatments was in height for both species (Figure 2). Given the variability in height growth, we modeled height growth as a function of a range of topographic indices and their log-transformed equivalents. Models within $\Delta AIC_c = 2$ of the best ponderosa pine height growth model explained 49.8-53.1% of the variability in height growth (Table 5). The only predictor variable that was in all but one of the best models was log-transformed topographic wetness index and the untransformed predictor was present in the other model (Table 5). These results indicate that there is a strong positive relationship between height growth and areas on the landscape that accumulate water.

Table 5 The five linear models with the lowest ΔAICc for ponderosa pine three-year growth using topographic predictor variables.

	Model 1	Model 2	Model 3	Model 4	Model 5
Intercept	5.36	54730.36	-18.65	53342.6	5.08
Elevation	-29.75	-	-29.01	-	-29.24
Heat load	-	-	-	-	-
Log(elevation	-	-74183.6	-	-72367.9	-
Log(heat load)	-	-	-	-	-
Log(topographic roughness)	-	-	-	-	-
Log(slope)	-	-	12.19	12.18	-
Log(topographic index)	-	74138.32	-	72327.73	-
Log(topographic wetness index)	26.33	26.50	28.50	28.65	25.71
Topographic roughness	-	-	-	-	9.86
Slope	1.13	1.13	-	-	-
Topographic index	61.95	-	60.39	-	62.13
Topographic wetness index	-	-	-	-	-
R2	0.519	0.518	0.516	0.515	0.514
AICc	305.90	306.04	306.20	306.32	306.37
ΔAICc	0	0.14	0.29	0.42	0.47

Models within $\Delta AIC_c = 2$ of the best Douglas-fir height growth model explained 59.2-68.7% of the variability in height growth (Table 6). Unlike ponderosa pine, topographic wetness index was not included in any of the best models. The five best models all included topographic roughness and slope or the log-transformed versions of those variables.

Table 6 The five linear models with the lowest ΔAICc for Douglas-fir three-year growth using topographic predictor variables.

	Model 1	Model 2	Model 3	Model 4	Model 5
Intercept	82.32	-46.08	-1.11	1622.30	1653.64
Elevation	-	-	-	-	-
Heat load	-	47.72	-	42.16	-
Log(elevation)	-	-	-	-	-
Log(heat load)	-	-	32.43	-	28.65
Log(topographic roughness)	-	-	-	783.71	780.07
Log(slope)	-65.26	-	-	-760.62	-756.54
Log(topographic index)	-	-	-	-	-
Log(topographic wetness)	-	-	-	-	-
Topographic roughness	58.87	467.56	464.40	-	-
Slope	-	-51.65	-51.23	-	-
Topographic index	-	-	-	-	-
Topographic wetness	-	-	-	-	-
R2	.591	0.687	0.680	0.673	0.665
AICc	103.63	103.71	104.05	104.44	104.81
ΔAICc	0	0.08	0.42	0.80	1.18

The results from the planting experiment suggest that we can improve the probability of survival by leveraging portions of the landscape where roughness is high and that are likely to accumulate water. We did not find that the cover of New Mexico locust was sufficient to buffer the microclimate and increase seedling survival. However, New Mexico locust has a lower specific leaf area than the other common post-fire shrub in this system, Gambel oak, and quantify the influence of Gambel oak on the below-shrub microclimate warrants investigation (Figure 3, Krofcheck et al. 2019).

Figure 3: *Quercus gambelii* (left) and *Robinia neomexicana* (right) leaves exhibiting morphology typical for each species in the Jemez Mountains. Figure from Krofcheck et al. (2019).

The same predictors, topographic wetness index for ponderosa pine and topographic roughness index for Douglas-fir were important for modeling height growth. The survival and height growth results combined suggest that by leveraging locations in the post-burn environment that have high topographic roughness and accumulate water could facilitate establishing patches of tree regeneration that will serve as seed sources for surrounding areas.

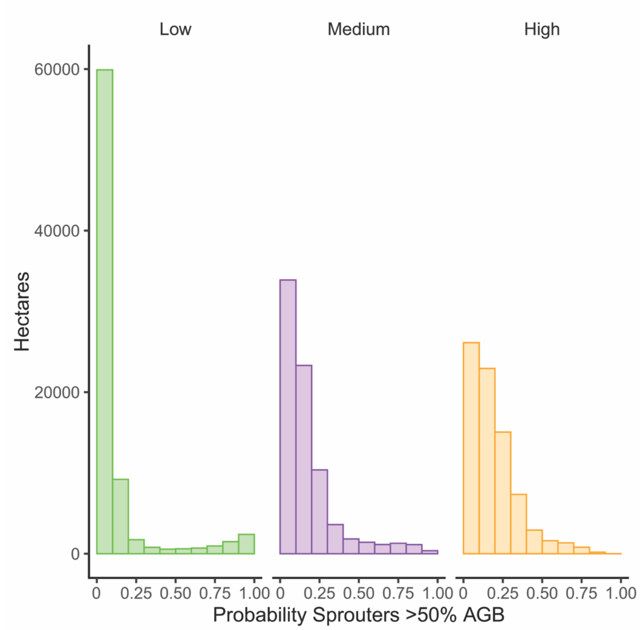

Figure 4: The distribution of probability that aboveground biomass of aspen, Gambel oak, and alligator juniper is greater than 50% of the total biomass for the low, medium, and high fire probability scenarios at the end of the simulation period. Figure from Keyser et al. (2020).

In this fire-prone system, seedling survival and growth is only part of the challenge for re-establishing forest following stand-replacing wildfire. We evaluated the effect of fire probability on the landscape successional trajectory for the Las Conchas Fire footprint. The three scenarios included a low probability of fire based on recent empirical fire occurrence data, a high probability of fire based on the historic mean fire return interval for this system, and a medium probability based on the average of the low and high values. Our results show that the successional development of the system is sensitive to the frequency of fire events, with the probability that the majority of the aboveground biomass is composed of resprouting species increasing substantially as the probability of fire increased (Figure 4, Keyser et al. 2020). Increasing fire frequency reduced the proportion of the landscape that was occupied by young conifer (primarily ponderosa pine and Douglas-fir) and by young pinyon-juniper (Figure 5, Keyser et al. 2020). These results suggest that, following high-severity wildfire, an increase in the probability of fire could reinforce the conversion to shrub and grassland, impeding the successional development of forest.

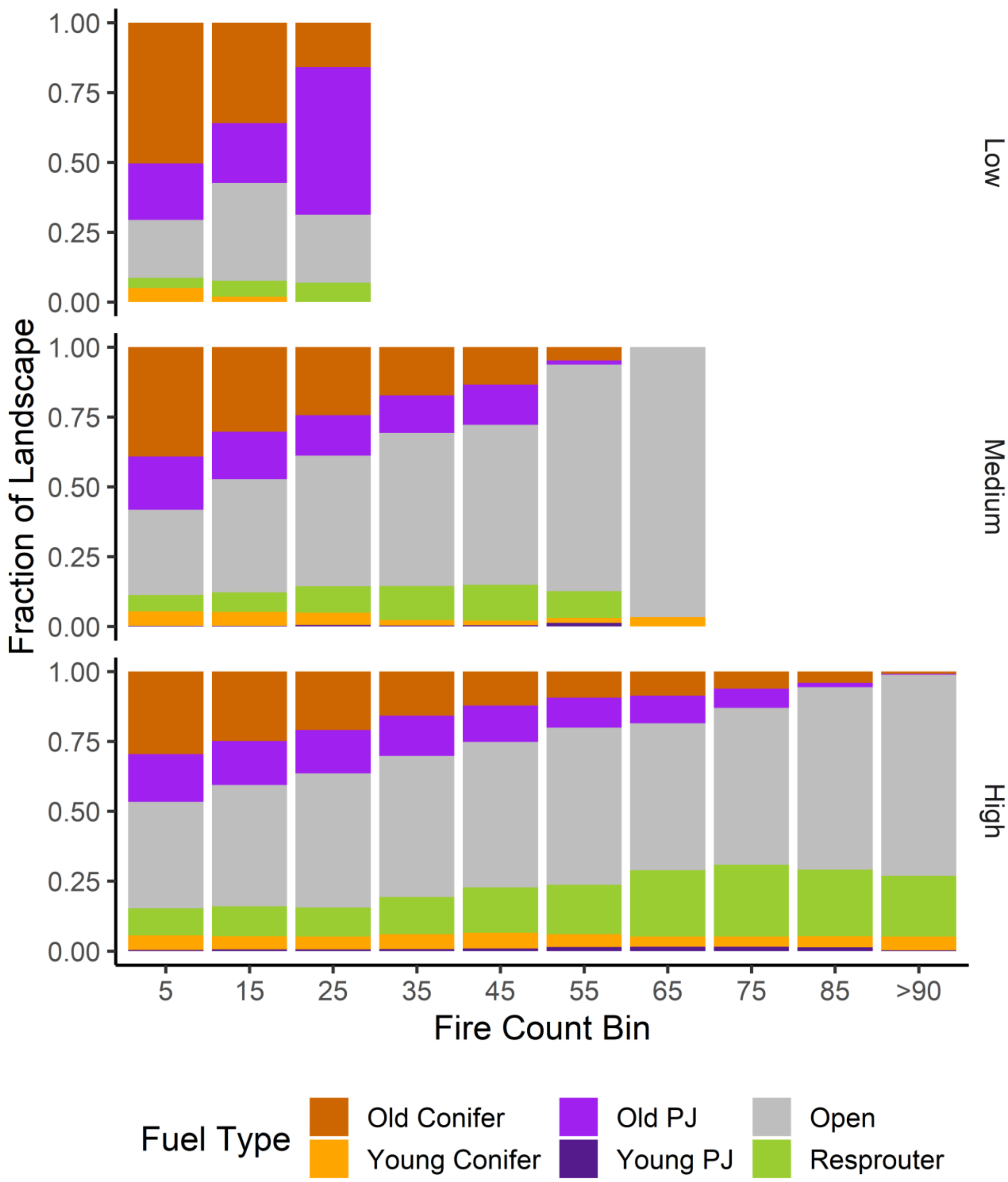

Figure 5: The distribution of fuel types across the landscape by the number of simulated fire events for the low, medium, and high fire probability scenarios. The labels identify the midpoints of fire count bins. The PJ bin includes the fuel type for pinyon-juniper; the conifer bin includes all other conifer fuels; the resprouter bin includes aspen and Gambel oak. The young conifer (≤ 11 years) and young PJ (≤ 21 years) correspond to the youngest group in each fuel type. The open fuel type is the equivalent of an herbaceous fuel layer and can include seedlings if present on the site. Figure from Keyser et al. (2020).

Conclusions and Implications for Management and Future Research

The interaction between high-severity fire and ongoing climate change is increasing the probability of conversion from forest to non-forest vegetation types (Coop et al. 2020). This is in part driven by long distances to seed source and hotter, drier conditions following stand-replacing wildfire. Planted seedling survival can overcome the dispersal distance limitation, but planted seedlings will also be challenged by the hotter, drier post-fire conditions. Our results indicate that we can leverage landscape position to increase the probability of survival and height growth. Accomplishing higher planted seedling survival and faster height growth for ponderosa pine and Douglas-fir in the southwestern US will require targeting areas of increased topographic wetness and increased topographic roughness. Targeting post-fire planting efforts to these areas will help increase the chance that we are able to establish tree clusters in large high-severity burn patches that can serve as seed sources once they mature.

The challenge to achieving establishment of tree clusters in large high-severity patches will be the chance that subsequent fire causes tree mortality. Our simulation results suggest that establishing trees will require at least ten fire-free years after planting. However, as our height growth results indicate, the fire-free period required to ensure seedling survival is going to vary as a function of how quickly seedlings grow at a given location. Given the low seedling survival rates in southwestern post-fire landscapes, our results suggest that rather than widespread planting efforts across all formerly forested areas, planting should be focused in topographic positions that are likely to increase both the probability of survival and growth. Given the sensitivity of young trees to fire, establishing tree clusters in large high-severity burn patches will likely require efforts to exclude fire from planted areas until the trees are of sufficient size to survive. The size required to survive fire will in-part be determined by the surrounding fuels, with larger sizes being required to survive fire in shrub fuels than in herbaceous fuels.

Our research points toward two areas in need of additional research: 1) a better understanding of how shrubs modify microclimate and 2) the influence of soil characteristics on seedling survival and growth. While we did not find an influence of New Mexico locust on planted seedling survival, the different architecture of other species, such as Gambel oak, may be more influential for modifying microclimate and influencing survival rates (Krofcheck et al. 2019). This will require both a characterization of the below canopy environment and experimental planting to determine if the survival rates are higher. The other aspect is the role of soils on planted seedling survival and growth. We did not control for soil type in our study, but differences in survival and growth between some of our planting locations that happened to be on pumice derived soils indicate that the influence of soil type is worthy of additional investigation. Given the positive correlation between topographic wetness index and ponderosa pine survival, soil water holding capacity could be an important predictor variable.

Literature Cited

Bartoń K. 2020. MuMin: Multi-model inference. R package v1.43.17. https://CRAN.R-project.org/package=MuMIn.

Collins BM, Roller GB. 2013. Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology 28:1801-1813.

Coop JD, Parks SA, McClernan SR, et al. 2016. Influences of prior wildfires on vegetation response to subsequent fire in a reburned southwestern landscape. Ecological Applications, 26:346-354.

Coop JD, Parks SA, Stevens-Rumann CS, et al. 2020. Wildfire-driven forest conversions in western North American landscapes. BioScience, 70:659-673.

Coppoletta M, Merriam KE, Collins BM. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications, 26:686-699.

de Bruijn A, Gustafson EJ, Sturtevant BR, et al. 2014. Toward more robust projections of forest landscape dynamics under novel environmental conditions: embedding PnET within LANDIS-II. Ecological Modelling, 287:44-57.

Finney MA. 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research 32:1420-1424.

Geroy IJ, Gribb MM, Marshall HP, Chandler DG, Benner SG, Mcnamara JP. 2011. Aspect influences on soil water retention and storage. Hydrological Processes 25: 3836–3842.

Gomez-Aparicio L, Gomez JM, Zamor R, Boettinger JL. 2005. Canopy vs. soil effects of shrubs facilitating tree seedlings in Medditerranean montane ecosystems. Journal of Vegetation Science 16:191-198.

Holmgren M, Scheffer M, Huston MA. 1997. The interplay of facilitation and competition in plant communities. Ecology 78:1966-1975.

Keyser AR, Krofcheck DJ, Remy CC, et al. 2020. Simulated increases in fire activity reinforce shrub conversion in a southwestern US forest. Ecosystems, doi: 10.1007/s10021-020-00498-4.

Krofcheck DJ, Litvak ME, Hurteau MD. 2019. Allometric relationships for *Quercus gambelii* and *Robinia neomexicana* for biomass estimation following disturbance. Ecopshere, 10:e02905.

McGinnis TW, Keeley JE, Stephens SL, Roller GB. 2010. Fuel buildup and potential fire behavior after stand-replacing fires, logging fire-killed trees and herbicide shrub removal in Sierra Nevada forests. Forest Ecology and Management 260:22-35.

Miller JD, Safford HD, Crimmins M, et al. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade mountains, California and Nevada, USA. Ecosystems 12:16-32.

Parks SA, Holsinger LM, Miller C, et al. 2015. Wildand fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecological Applications 25:1478-1792.

Plamboeck AH, North M, Dawson TE. 2008. Conifer seedling survival under closed-canopy and manzanita patches in the Sierra Nevada. Madrono 55:193-203.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. https://www.R-project.org/

Remy CC, Krofcheck DJ, Keyser AR, et al. 2019. Integrating species-specific information in models improves regional projections under climate change. Geophysical Research Letters, 46:6554-6562.

Savage M, Nystrom Mast J, Feddema JJ. 2013. Double whammy: high-severity fire and drought in ponderosa pine forests of the southwest. Canadian Journal of Forest Research 43:570-583.

Scheller RM, Hua D, Bolstad PV, et al. 2011. The effects of forest harvest intensity in combination with wind disturbance on carbon dynamics in Lake States Mesic Forests. Ecological Modelling 222:144-153.

Singleton MP:, Thode AE, Sanchez Meador AJ, Iniguez JM. 2019. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. Forest Ecology and Management, 433:709-719.

Sthultz CM, Gehring CA, Whitham TG. 2007. Shifts from competition to facilitation between a foundation tree and a pioneer shrub across spatial and temporal scales in a semiarid woodland. New Phytologist 173:135-145.

Westerling AL, Bryant BP, Preisler HK, et al. 2011. Climate change and growth scenarios for California wildfire. Climatic Change 109(Suppl 1): S445-S463.

Appendix A

PI:
Matthew Hurteau
University of New Mexico
MSC03 2020
Albuquerque, NM 87104
505-277-0863
mhurteau@unm.edu

Co-PIs:
Dan Krofcheck
University of New Mexico
MSC03 2020
Albuquerque, NM 87104
krofcheck@gmail.com

Craig Allen
USGS New Mexico Landscapes Field Station
15 Entrance Rd
Los Alamos, NM 87544
505-672-3861 ext 720
craig_allen@usgs.gov

Appendix B

Articles in peer-reviewed journals:

Keyser AR, Krofcheck DJ, Remy CC, Allen CD, Hurteau MD. 2020. Simulated increases in fire activity reinforce shrub conversion in a southwestern US forest. Ecosystems, doi: 10.1007/s10021-020-00498-4.

Krofcheck DJ, Litvak ME, Hurteau MD. 2019. Allometric relationships for *Quercus gambelii* and *Robinia neomexicana* for biomass estimation following disturbance. Ecopshere, 10:e02905.

Remy CC, Krofcheck DJ, Keyser AR, Litvak ME, Collins SL, Hurteau MD. 2019. Integrating species-specific information in models improves regional projections under climate change. Geophysical Research Letters, 46:6554-6562.

Hiers JK, Hurteau MD, van Mantgem P, Hall JA, Rappold AG, Steblein P, Teensma PDA, Brunson E, Lahm P, Zupko M. In review. A framework for assessing wildland fire-societal value interactions. Fire Ecology.

Remy CC, Keyser AR, Krofcheck DJ, Litvak ME, Hurteau MD. In review. Future fire-driven landscape changes along a southwestern US elevation gradient. Climatic Change.

Presentations/webinars/other outreach/science delivery materials:

Research Brief: Fire reinforces forest conversion to shrubland.

Crockett, J., M.D. Hurteau. Quantifying the physical controls on post-wildfire vegetation establishment in the Southwestern US. 8^{th} International Fire Ecology and Management Congress.

Keyser, A., D.J. Krofcheck, C.C. Remy[‡], M. Hurteau. Quantifying the influence of fire probability on post-fire vegetation development in the Southwestern US. 8^{th} International Fire Ecology and Management Congress.

Hurteau, M.D. (invited). Quantifying abiotic and biotic controls on seedling survival in a post-burn environment. 2019 meeting of the US International Association of Landscape Ecology.

Hurteau, M.D. (Keynote) Wildfire and carbon management. Wildfire and Climate Change in the Kootenays Conference, June 26-28, 2018, Nelson, British Columbia.